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Article

Evaluating Plantation Forest vs. Natural Forest Regeneration for Biodiversity Enhancement in Hong Kong

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Abstract: Global trends predict a continuous increase in the proportion of forest occupied by plantations up to the end of the 21st century, while a dramatic loss of biodiversity is foreseen as a result of anthropogenic exploitation and climate change. This study compares the role and performance of plantation policies in Hong Kong, with natural regeneration of secondary forest, using detailed spatio-temporal data extracted from a previous study. The study extends over a 70-year period from 1945 to 2014 using aerial photographs and satellite images of five time periods to document spatio-temporal trends in plantation forestry and natural forest succession. Field data on species richness and woody biomass at different stages of forest succession are compared with available data from plantations in the same study area. Results indicate that plantation forests support relatively few native species in the understory, with much lower species richness than naturally regenerated forest, even after 6 to 7 decades. Time-sequential maps of habitat change show that natural forest succession from barren grassy hillsides, progressed at an annual rate of 7.8%, from only 0.2% of the landscape post WWII, to over 37% today. Plantation forestry on the other hand has been less successful, and has even acted as a barrier to natural forest regeneration, as mono-cultural plantations from the late 1960s to 1980s are still plantations today, whereas other similar areas have succeeded naturally to forest. The theory of plantations acting as a nurse crop for a woody native understory is not supported, as *Pinus massoniana* plantations, destroyed by two deadly nematodes during the 1970s, apparently had no woody understory, as they were seen to have reverted to grassland in 1989 and are still mainly grassland today.

Keywords: plantations; biodiversity; secondary forest; Hong Kong



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1. Introduction

Only 3% of the world's forests are now primary, with a decline of over 8% since 1990. However, in east Asia only 6% of forest is primary forest [1]. Although some of the primary forests cleared for farming are regenerating naturally following abandonment, these naturally regenerating forests have also declined worldwide by approximately 8% since 1990. Naturally regenerating or secondary forests differ from primary forests in their vastly simplified structure and species composition, but generally comprise a range of native woody species. On the other hand, planted forests, or tree plantations, in east Asia (excluding oil palm) currently represent 36% of forest compared with 7% worldwide [2]. Global trends predict a continuous increase in the proportion of forest occupied by plantations up to the end of the 21st century [2,3].

Due to the importance of forest ecosystems in supplying a range of essential ecosystem services for human life, it is necessary to evaluate the ability of forest plantations to provide these vital support functions. However, their ability to do so may be compromised by the use of non-native and fast-growing species [4], the use of which increased worldwide by 29% between 1990 and 2020, and by 138% in east Asia [2]. East Asia's large increase

in forest plantations is largely due to massive afforestation programs in China during the 2000–2010 decade [5], which mainly comprised fruit trees, rubber and eucalyptus. Although such monoculture plantations may serve some of their intended purposes such as carbon storage, economic productivity and control of soil erosion, these often come at the expense of other ecological functions [5]. Additionally, the use of single or few species makes them less resistant than biodiverse natural forests to external disturbances such as fire, disease and climatic fluctuations [6]. Biodiverse forests contain a wide range of species enabling them to perform wide ranging ecological functions and adapt to changing ecological conditions. Furthermore, studies have demonstrated a positive relationship between species richness in tropical forests and carbon storage [7–9].

There are two viewpoints about the role of plantation forestry's ability to restore ecological functions and biodiversity. Firstly, that it is detrimental, and at best, its use represents a trade-off between socio-economic and ecological goals [5,6,10,11]. Secondly plantations have sometimes been recommended as beneficial, as they can assist the early stages of succession to a stable and biodiverse forest ecosystem [12–17].

Especially where land has become severely degraded, plantations are sometimes viewed to have catalytic effects, by providing an understory, suitable microclimate and build-up of a litter and humus layer, for native species to regenerate naturally [12,13,18]. Piironen et al. (2016) [17], for example, suggest that tree communities in plantations in the Afro-tropics are moving towards old growth forest due to plantation trees suppressing persistent grass invasion following reforestation. In tropical and sub-tropical Australia, plantations, whether of single or mixed species, were considered to have strongly positive consequences for restoring biodiversity in heavily cleared landscapes [19]. However, this viewpoint is countered by another study in tropical Australia, where dung beetles, considered to be useful environmental indicators, showed lower species diversity and numbers, in plantations than in native forest [20]. Furthermore, in a summary of research papers on south-east Asia, where plantation forests have increased by approximately 1.3% annually since 2010 [1], all types of non-oil palm plantations were shown to be inferior to naturally regenerating forest in species richness and abundance [21]. The lower richness and abundance were attributed to the lack of preferred microhabitats and floristic diversity causing reduced insect diversity, as well as lack of perennial fruits and food sources for birds. Nevertheless, the authors [21] do agree with other studies that plantation provides a better biodiversity alternative to other modified non-forest landscapes [12,16].

The question of whether plantations are detrimental or beneficial in forest regeneration was addressed by a survey of regenerating forests worldwide [22] which suggests that the previous land use before abandonment largely determines if forest could recover naturally or would require interventions. Overall, biological processes recovered slower following agriculture than logging or mining, and biodiversity recovery in tropical forest was slower than in temperate forests. Although the objectives of plantation establishment vary widely in different world regions, evaluation of their ecological and environmental effects does not always consider these objectives [16]. If the primary objective is restoration of a biodiverse and stable ecosystem, the high cost of preparation, planting and nurturing of tree seedlings, as well as subsequent fire protection, may make plantation an inferior alternative to natural forest regeneration if the latter is feasible.

In the highly degraded hillsides of Hong Kong, planting programmes have been in place since the 1870s, and Hong Kong is thought to be the earliest example of plantation forestry in the tropics [13]. The main objectives have shifted from health and aesthetic considerations in the 19th century, to preventing soil erosion and watershed protection in the 20th century, and to ecological restoration today [13]. This has paralleled global concerns surrounding loss of biodiversity in all continents due to habitat loss accompanied by climate change [23]. Earlier work in Hong Kong by Lee et al. [24] concludes that 'few studies have compared plantations with naturally regenerated forest of similar age. Natural succession with little or no intervention might have been a more effective rehabilitation method'. More recent work in southern China and Taiwan appears to confirm this as,

for example, Liu et al. [25] observed less than half the number of tree species seedlings in the understory of plantation compared to natural hardwood forest and Su et al. [26] reported that, fir plantations showed low regeneration in terms of understory shrub cover and diversity compared to natural secondary forest.

The availability of a seven-decade long archive of aerial photographs and satellite images enables an evaluation of the relationships between plantations and natural succession in Hong Kong's tropical secondary forests. The objectives of this study are therefore to evaluate the benefits of plantations in the process of natural forest regeneration in Hong Kong, by comparing the outcomes, of naturally regenerated forest, with areas which have undergone planting, over the last seven decades. Data for the naturally regenerating forested areas are derived from our previous studies [27,28], and data for plantations are taken from Lee et al. [24]. This paper uses the term 'regeneration' to refer to the process of natural forest succession from grassland to secondary forest, on land where forest had existed at some stage in the past.

2. The Study Area

Hong Kong's landscape is rugged and mountainous, with convex slopes rising from sea level to almost 1000 m, within a few kilometres of the coast (Figure 1). Scarcity of flat land has confined urban development to only 20% of the land area, leaving over 80% relatively undeveloped, supporting grassland, shrubland and forest. In 1976, 40% of Hong Kong's land area was designated for protection in Country Parks and these are administered by the Agriculture, Fisheries and Conservation Department (AFCD). Two of Hong Kong's 24 Country Parks, Tai Mo Shan and Shing Mun in the New Territories, totalling almost 30 km² were selected for the study. Climatically, Hong Kong experiences cool dry winters and hot humid summers. Its position, at latitude 22.3 °N, on the edge of the tropics, supports a highly diverse flora and fauna belonging to both tropical and sub-tropical genotypes. However, the flora is recognised as tropical except for high altitude areas where some sub-tropical plants are reported. The main climatic limitations to plants are occasional frost events, which limit tropical species to low altitudes, and typhoons. The fauna is recognised as tropical [29].

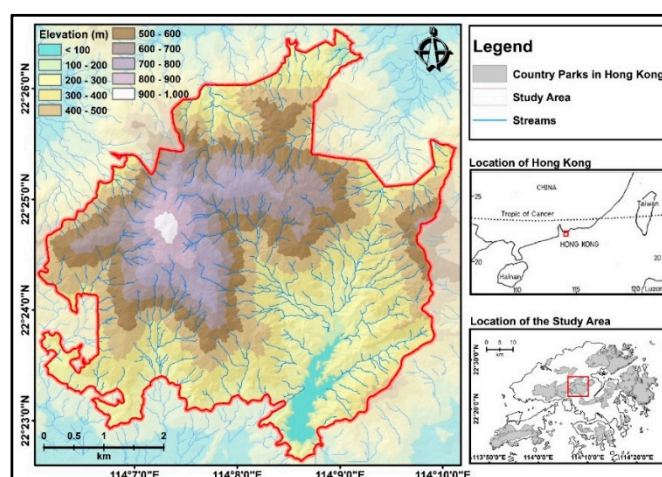


Figure 1. Location of study area within Hong Kong and China.

Corlett (1999) [13] states that it is likely Hong Kong's original tropical evergreen broadleaf forests were removed in the 15th to 17th centuries, and planting by the colonial government began around 1871 to make the, then barren, hillsides more attractive. Plantation efforts continued over the next 8 decades with a variety of species dominated by the native *Pinus massoniana*, as well as *Castanopsis fissa* and other exotic broadleaves including several species of *Eucalyptus*, *Lophostemon confertus*, *Acacia confusa*, *Leucaena leucocephala* and *Melaleuca quiquinervia*. Following almost complete devastation of Hong Kong's wooded

areas during the blockade years of WWII when trees were cut for timber and fuel, massive plantation forestry took place. The objectives were to control soil erosion on the barren hill-sides, to improve water catchment functions, and for production forestry. Similar species, dominated by *Pinus massoniana*, were used as during pre-WWII [13]. Only after 1965 was conservation of wildlife seen as an objective of Hong Kong's forestry policy, along with countryside recreation and protecting the water catchment areas. Thereafter, due to their fire susceptibility, *Pinus massoniana* and *Eucalyptus spp.* were less favoured, and a few easily propagating exotics have predominated, including *Lophostemon confertus*, *Pinus elliotii* and *Acacia confusa* [24].

3. Methods

The study compares the outcomes of forest plantation policies in Hong Kong, with those of natural forest succession, using data from two sources. Firstly, the present authors [27,28] utilised an image archive covering five time periods between 1945 and 2014, to track and map changes in land cover and habitat development (including both natural vegetation and plantation development) over the 7-decade period. The image archive is described in Section 3.1 below.

The second source was a study of plantations by Lee et al. (2005) [24] of the same study area in Hong Kong, including inventories of species richness and structural characteristics of plantations. Lee et al. used sequential aerial photographs to determine plantation ages and conducted plot surveys on species richness and basal area, by plantation age and species type. Figure 2 (flowchart) summarises the steps of the study.

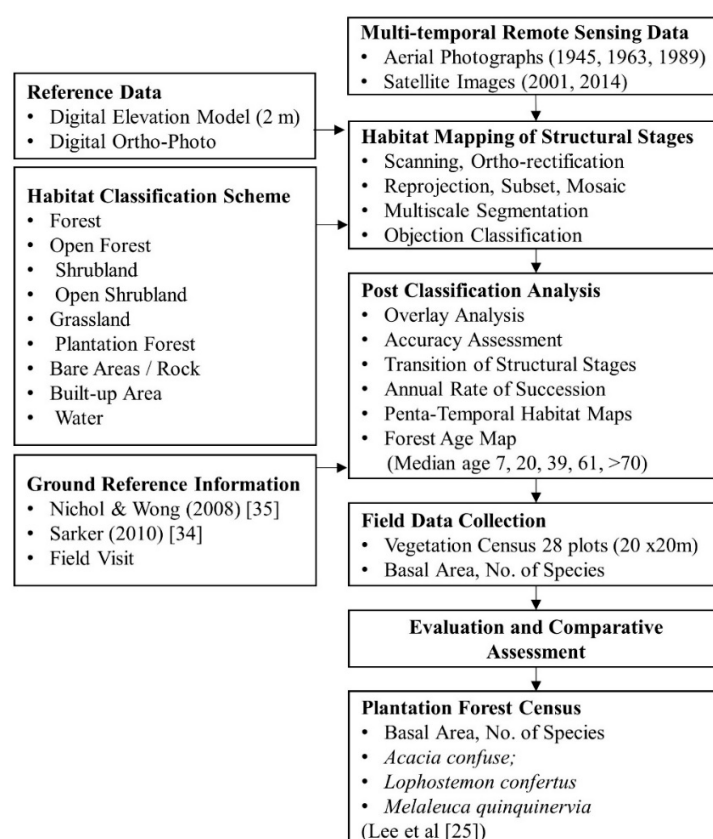


Figure 2. Flow chart indicating steps by step procedures of the study.

3.1. Remote Sensing Data

The image archive covered five time periods between 1945 and 2014, to track and map changes in land cover and habitat development over the 7-decade period. These comprised aerial photographs of 1945 at 1:40,000 scale, 1963 at 1:14,000 and 1989 at 1:20,000 scale,

totalling 43 aerial photographs (18 for 1945, 12 for 1963 and 13 for 1989). For the two most recent dates, we obtained high resolution IKONOS satellite images of 2001 with 1 m spatial resolution, and for 2014, WorldView-2 images with 0.5 m spatial resolution.

For accurate comparisons over time, it was necessary to register all five sets of images to the same digital coordinate system. For this, the individual air photo prints were converted to orthophotos by scanning at a resolution of 1200 dpi, followed by orthorectification using 15 to 20 GCPs per photo. The X, Y and Z coordinates of the GCPs were obtained from a 0.5 m digital orthophoto of Hong Kong and a 2 m Digital Elevation Model (DEM). The multiple orthophotos were then clipped and mosaicked to form unified orthoimages. The satellite images were also converted to orthophotos using RPC (Rational Polynomial Coefficient) files, the Hong Kong orthophoto and the DEM. All the data sets were then re-projected into the Hong Kong 1980 Grid system. Co-registration of all datasets to the digital orthophoto produced an RMS error of 1 to 5 m, depending on scale and resolution of the dataset.

3.2. Habitat Classification

For vegetation classification, the study adopted an ‘a priori’ scheme from a previous habitat mapping project [30], as it was based on structural characteristics interpretable from remote sensing images. This scheme itself, is derived from a standard system, the Land Cover Classification System (LCCS) [31] (Table 1). The LCCS was devised by FAO and UNEP, to standardize data on land use/land cover, and is universally applicable [32]. The classes are easily identifiable on remote sensing images and are mutually exclusive and unambiguous.

Table 1. Description of habitat classes and structural stages of vegetation.

Habitat Class (Structural Stage)	Description (Based on LCCS Definitions)
Forest	Broadleaved, evergreen trees, with >60–70% canopy cover
Open Forest	Tree/shrub/grass mosaic: open tree canopy (15–60%) with shrubland and/or grassland
Shrubland	Broadleaved, evergreen shrubland, with closed canopy >60–70%
Open Shrubland (Shrubby Grassland)	Shrub/grass mosaic: open canopy of shrubs (>15–60%) with grassland
Grassland	Ground story of grasses (>70%) as dominant vegetation form as well as <15% shrubs/trees
Plantation Forest	Monocultural plantation stands: <i>Lophostemon confertus</i> , <i>Acacia confusa</i> , <i>Melaleuca quinquenervia</i> , as well as mixed spp.

3.3. Multi-Scale Image Segmentation and Habitat Classification

Delineating habitat boundaries on the different types and scales of imagery from the five different time periods manually, is necessarily subjective. Therefore, an automated, object-based method was devised, which uses the spatial and tonal signatures of pixel groupings to segment the image into multi-pixel image objects using the multi-resolution segmentation approach. These defined objects maximize the variability between different objects and minimize that within objects, considering the expected size and scale of objects. Thus, the segmentation process represents the automated digitizing of target boundaries. The segments, representing habitat patches were then assigned to a habitat type, while taking into account their respective spatial scales.

Habitat patches can be of any size and can grow or contract over time. Patchiness can also occur at finer scale, as each patch has internal structure, as well as at broader scale when similar patches are amalgamated. In this study, multi resolution segmentation was executed at three sizes of Minimum Mapping Unit: 1000 m² (for large patches of grassland, forest and plantations), 500 m² (for open shrubland and shrubland) and 50 m² (for isolated forest and shrubland patches). All the image interpretation, to classify the segments, was performed by a single expert analyst to minimize interpretation errors. Plantation stands were recognised by homogeneity of tonal pattern, as well as multi-temporal analysis, as distinct plantation patterns were easier to recognize in the early stage. For simplicity in change analysis, a single class of plantation forest was created by merging the four sub-

classes of plantation species *Lophostemon confertus*, *Acacia confusa*, *Melaleuca quinquenervia* and mixed plantation. The mapping was validated using 352 field surveyed GPS points and 215 additional check points on very high-resolution colour air photos, from a previous habitat mapping project [33,34].

The allocation of image objects delineated by multi-scale segmentation to a particular structural class was conducted manually, and is necessarily subjective. However, observation at multiple time steps was able to improve the mapping, as successional stages usually proceed logically. For example, it was difficult to distinguish between stands of mature plantation and forest on a particular image date, but reference to earlier growth stages (as young plantations are easy to identify and tend to precede mature plantations, and shrubland precedes forest) helped to reduce the subjectivity. Forest and plantation were identified with 98% and 100% accuracy respectively, and overall mapping accuracy exceeded 92%. The percentage rate of change for each class was calculated by using Equation (1) [35] which is not sensitive to difference in periods and provides a more meaningful estimation of annual change rate.

$$R = \left(\frac{1}{T_2 - T_1} \right) \times \left(\ln \frac{a_2}{a_1} \right) \times 100 \quad (1)$$

where R is the rate of change (% per year), and a_1 and a_2 represent area corresponding to earlier time, T_1 , and late time, T_2 , respectively.

3.4. Field Data Collection

Field survey in the naturally regenerating (secondary) forest was conducted in 28 plots of 20 by 20 m (400 m²) from March 2015 to May 2016, to document and quantify the species composition and stand density of forest plots at different stages of succession [36]. This included species richness, i.e., the total number of species occurring at each successional stage and basal area as a surrogate for total forest biomass. Plot selection considered the heterogeneity of the landscape and accessibility, to include a range of age classes, altitudes, aspects and slope. All trees having a diameter at breast height (DBH) ≥ 1 cm were identified and measured and the basal area for each plot was recorded. The age of a plot was estimated from sequential interpretation of vegetation cover, and plots were grouped into five median age classes: 7, 20, 39, 61 and >70 years. In total, the plots occupied 1.12 hectares at elevations between 205 m and 822 m. A total of 8575 plants belonging to 229 species and 63 families were tagged, measured, identified and recorded, requiring 2800 man-hours. The field data collected by forest age class enabled comparison between plantation forestry and natural forest succession, in their progress toward a biodiverse and stable ecosystem, over the several decades of the study. To enable comparisons with the ages of plantation, plot data for secondary forest were interpolated between the surveyed years using the annual rate of change and 2-sample t -test was applied to determine statistical significance of the differences.

4. Results

4.1. Landscape Change Over Time

Table 2 shows the area in hectares of each habitat type at each of the five mapped years, for the four major habitat classes and Figure 3 shows the habitat map at five stages of succession. Over the whole 70-year period, the area of forest increased at a rapid rate of 7.8% annually.

In 1945, grassland comprised 79% of the landscape, the remainder being barren (12%), shrubland (7%) and only 0.17% of forest (totalling 5 ha.), which was mainly confined to Fung Shui woodlots adjacent to villages, and remote ravines. During the 18-year period from 1945 to 1962, natural succession was very slow, little change occurred in the landscape, and the area of forest only increased to 1.7% of the landscape, with over half of this being open forest. However, large-scale post-war afforestation programmes were mounted following WWII, and plantations increased from zero in 1945 to 15% (432 ha) by 1963,

concentrating in the south-west of the study area (Figure 3). The native *Pinus massoniana* was the dominant species in this area.

Table 2. Area statistics (ha. and %) for habitat classes 1945 to 2014 [28], where Forest class includes both open and closed canopy forest, and Shrubland class includes both open and closed canopy shrubland. Built areas and bare areas are not included.

Habitat Class	1945		1963		1989		2001		2014	
	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)
Forest	4.82	0.17	19.85	0.71	140.03	5.01	579.04	20.70	1019.23	36.44
Open Forest	0.22	0.01	27.69	0.99	38.09	1.36	108.32	3.87	27.54	0.98
Shrubland	24.03	0.86	175.05	6.26	464.18	16.59	418.70	14.97	650.43	23.25
Open Shrubland	170.99	6.11	203.57	7.28	245.27	8.77	261.49	9.35	421.29	15.06
Grassland	2203.17	78.74	1805.62	64.53	1479.82	52.90	969.92	34.68	219.92	7.86
Bare Area	336.95	12.04	71.76	2.56	52.57	1.88	52.16	1.86	51.23	1.83
Built-up Area	0.41	0.01	5.21	0.19	8.80	0.31	12.51	0.45	12.51	0.45
Water	57.21	2.04	57.21	2.04	57.21	2.05	57.21	2.05	57.21	2.05
Plantation Forest	0.00	0.00	431.85	15.43	311.20	11.13	337.81	12.08	337.81	12.08
Total (ha)	2798									

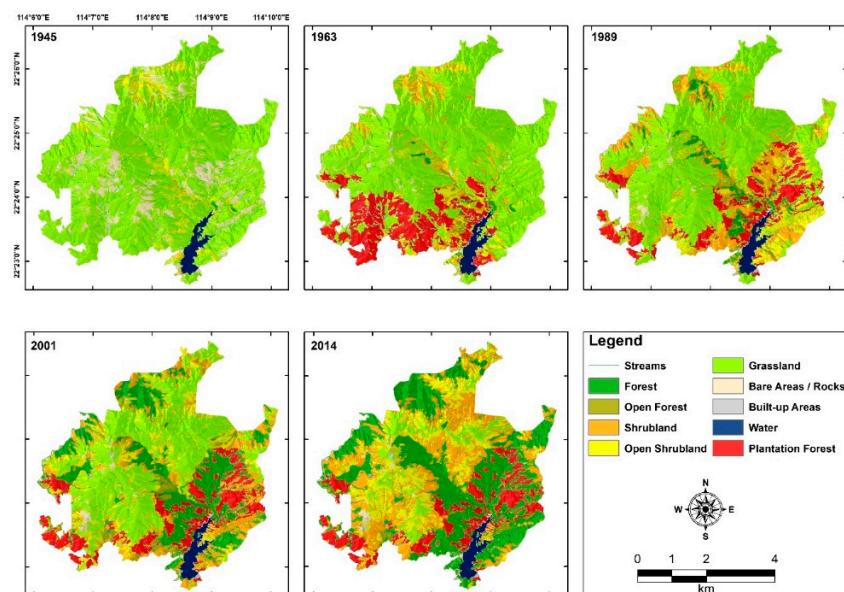


Figure 3. Land cover classes at five time intervals from image interpretation.

The 26-year period 1963 to 1989, saw a greater increase in the percentage of the landscape occupied by secondary forest, from 0.17% in 1963 to 6.4% in 1989. Thus, even after 44 years of natural succession following WWII, forest still only occupied 6.4% of the landscape, as the transition from grasslands through a shrubland stage, into open forest, then closed forest was slow. It is unfortunate therefore that the *Pinus massoniana* plantations established in the earlier period in the south-west, were destroyed by two diseases: the pine needle scale and a pinewood nematode [13], reducing the plantation areas from 15% in 1962 to 11% in 1989. The areas occupied by the plantations were seen to have immediately reverted to grassland by 1989, suggesting that no natural invasion by a tree or shrub understory had taken place during the plantation period.

The highest annual rate of increase in forested area, of 11% per year, was seen in the 12-year period from 1989 to 2001. During this period the proportion of the landscape occupied by secondary forest more than tripled, from 6.4% to 24.5% (from 179 to 787 ha.). This involved an increase along elevation as shrublands spread upwards from valley bottoms and transitioned into forest. Plantation areas increased only slightly, by 1%, to

occupy approximately 12% of the landscape by 2001, as those areas of mixed species not destroyed by disease persisted. It is notable that the areas of the destroyed *P. massoniana* plantations were still, by 2001 largely covered by grassland.

Although the highest rate of change in forest was in the period up to 2001, the highest annual gain in forested area was from 2001 to 2014 (Table 2). By 2014, secondary forest occupied over 37% of the landscape, and covered most valley bottoms and steep-sided slopes, except for those areas covered by plantation forests, mainly in the Shing Mun Reservoir catchment (Figure 3). These plantations continued to occupy 12% of the landscape, with no change in the 13 years since 2001. By 2014, the area of shrubland occupied a similar percentage of the landscape to secondary forest, at 38% (orange and yellow on Figure 3), and grassland had now diminished to 8.5%, almost exclusively on mountain summits (pale green on Figure 3). It is notable that the areas occupied by the *Pinus massoniana* plantations established in the south-west of the study area post-WWII and destroyed by disease in the 1970s, supported only grassland and open shrubland by 2014 (Figure 3).

4.2. Species Richness and Woody Biomass in Secondary Forest and Plantations

Field plots in the regenerating secondary forest, showed an average of 26, 33, 37, 38 and 28 tree species in plots with median ages of 7, 20, 39, 61 and 70 years. In contrast, Lee et al. (2005) [24], using the same sized (20 by 20 m) plots in plantation forests in Hong Kong, report an average of 15 tree species in *A. confusa* (median age 25 years), 11 in *L. confertus* (median age 33 years) and 25 in *M. quinquinervia* plantations (median age 45 years) (Table 3). Thus, even the youngest secondary forests contained more tree species than any of the plantations. Moreover, the forest plots contained more than double the number of species reported for plantation plots of similar age, except for the deciduous plantation species *M. quinquinervia*, although this still contains considerably fewer species than secondary forest of similar age. The *t*-test results suggest that *A. confusa* and *L. confertus* stands were significantly different from similar aged secondary forest, but that *M. quinquinervia* is less significant, which may be due to its age (45 years) or the species itself may support natural invasion. In terms of basal area (BA), which represents woody biomass, secondary forest has considerably higher BA in two out of three species of plantation, but lower than in the deciduous *M. quinquinervia* plantations. Interestingly, these had reached comparable biomass to 70-year old secondary forest after only 45 years. Thus, to some extent the results confirm the positive relationship between species richness and carbon storage observed in other tropical forests [7–9].

Table 3. Mean (SD) values of tree species and basal area (m²/ha) in 20 m² plots in secondary forest and plantations.

Age (Years)	n	Secondary Forest		Plantation Forest			2-Sample <i>t</i> -test between Species Richness		
		No of Species	Basal Area	Monoculture Plantation Stand	n	No of Species	Basal Area	t-Value	P-Value
7	5	26 (6.8)	25						
20	6	33 (11.53)	36						
25 *	6	34 (11.53)	37	<i>Acacia confusa</i>	4	15 (3.6)	18	3.77	0.009
33 *	8	36 (13.97)	39	<i>Lophostemon confertus</i>	4	11 (4.6)	25	4.59	0.001
39	8	37 (13.97)	40						
45 *	8	37 (13.97)	41	<i>Melaleuca quinquinervia</i>	4	25 (8.7)	61	1.82	0.102
61	5	38 (6.16)	43						
70	4	28 (10.72)	69						

* Plantation data reported by Lee et al., 2005 [24]. Plot ages and values for secondary forest refer to interpolated ages and values.

5. Discussion

Naturally regenerated secondary forest is shown to be superior to forest plantations in terms of species richness, with natural forest containing approximately twice the number of tree species as in plantations of the same age reported by Lee et al. (2005) [24] in Hong Kong. This is not to say that natural succession will continue accumulating species, and in the longer term, the limited species pool following centuries of deforestation and

loss of native fauna will limit the future succession [29]. Thus, intervention will probably be necessary to introduce some middle and late successional species.

There is also evidence [37,38] that forest plantations in Hong Kong are less able to withstand external disturbance, than native forest. In a severe typhoon in September 2018 [38], stands of exotic monoculture plantation (*Melaleuca quinquenervia*, *Lophostemon confertus* and *Acacia confusa*) were the most seriously damaged by the typhoon, with more than 25% decrease in vegetation vigour, represented by the Normalised Difference Vegetation Index (NDVI). The canopy was almost completely destroyed and as there was no generation of young understory trees to replace those lost, very little standing vegetation remained in the plantation patches. The canopy of these plantation patches comprises tree species not adapted to a typhoon climate, unlike those of secondary forest which appear more adapted to storms. The extensive typhoon damage in plantations is potentially double-edged, as the dead trees and debris following a typhoon provide enhanced fuel supply for grassland fires which occur frequently in the dry season. Plantations normally show low fire susceptibility due to their poor understory. From a sustainability viewpoint therefore, natural forest is clearly preferable to plantations, quite apart from the barrier they impose to natural forest regeneration, as seen from the time sequence analysis presented here.

During the period of greatest natural forest advance (1989 to 2001) the main barrier to advance was apparently the plantations established in the 1970s to 1980s, which impeded shrub colonisation and subsequent succession to forest. Even in the latest period 2001 to 2014, the plantations established earlier remain unchanged, and occupied 12% of the landscape, on valley and interfluve sites which elsewhere had undergone natural succession to forest. By 2014 natural forest occupied 37.4% of the landscape, and due to invasion of shrubland into grassland, shrublands occupied a similar proportion to forest i.e., 38% in 2014. Since the conversion of grassland to shrubland appears to be a much slower process than that of shrubland conversion to forest [28,39] this bodes well for the rapid succession of the remaining shrubland areas to forest, as the grassland to shrubland stage has been largely surpassed. The observation that the areas occupied by the *Pinus massoniana* plantations destroyed by disease in the 1970s, still supported only grassland and open shrubland by 2014, where other areas had succeeded naturally to forest, also suggests that plantations may impede natural succession. It counters arguments that plantations can promote understory native plant regeneration [40], or act as a nurse crop for diverse native seedlings [13]. It also counters arguments that plantations have strongly positive biodiversity benefits for restoration of heavily cleared landscapes [19]. Therefore, the observation of several studies that plantation provides a better biodiversity alternative to other modified non-forest landscapes [12,16,21] is not supported by our study.

The other main counter-argument for afforestation vs. natural regeneration in Hong Kong, is the considerable cost and effort required, which is difficult to estimate in monetary terms due to inflation over the decades. Corlett (1999) [13] describes huge labour costs over many decades starting in the 1870s including seed collection, sowing in the nursery and directly on hillsides, trials of different species, many of which failed, hand removal of parasites, application of chemical pesticides and fertilisers, thinning, brushwood cutting and maintaining firebreaks. During the 1960s to 1990s decades for example, Hong Kong's AFCD was hand-planting 300–350,000 seedlings a year. Even if the main objective for plantations is timber harvesting, the basal areas reported for plantations in our study area in Hong Kong [24] were lower than for secondary forest, for comparable aged stands.

Evidence from archived satellite images (Abbas et al., unpublished) indicates that frequent grassland fires during the 1980s, were a major impediment to shrub colonization in grasslands, which would explain the slow progress in the successional stage from grassland to shrubland noted above. It would also explain the faster progression from shrubland to forest, as shrubland is less susceptible to fire. An apparent decrease in hill fires observed on the satellite images during the 1990s (ibid) may result from more attention by the AFCD to fire control during the 1990s [13]. The latest figure reported here, for 37% of the landscape occupied by forest in 2014, is remarkable, in view of Corlett and

Turner's (1997) [41] statement that the 14% of the landscape occupied by forest (in 1997) was probably the largest area occupied by forest in Hong Kong for several centuries.

6. Conclusions

Over the 70-year period, plantation areas increased, then declined due to disease. Those areas where plantation species had been eliminated, were then completely lacking in woody species i.e., a woody understory of native species had not been established, and they then reverted to grassland. Although these events occurred in the 1970s, these areas remain as grassland today, whereas other areas which were not afforested, have now proceeded naturally to shrubland or secondary forest. Thus, planted forests have acted as a barrier to the establishment of a biodiverse natural forest ecosystem, although they may have satisfied the original objectives of forest policy in Hong Kong, of watershed protection and production forestry. Therefore, the plantations established in the 1960s have left a negative legacy for current biodiversity. Furthermore, the 12% of the landscape currently occupied by plantations not destroyed by disease during the 1970s, exhibit poor colonisation by native species and they appear to have prevented natural forest succession in the study area. This limited native colonisation of plantations may result from limited seed dispersal mechanisms, given the vastly reduced native fauna, especially those capable of dispersing large seeds. Lee et al. (2005) [24] state that only a few species of bird-dispersed shrubs were found in the understory of plantations.

The argument for plantations promoting native plant regeneration in the understory, or for plantations acting as a nurse crop, is not well supported by our findings. The limited lifespan of exotic plantation species, of 50 or 60 years, given increasing emphasis on conservation of biodiversity, is a major motivation for current policies of the AFCD, to artificially introduce native species into plantations. Given the high cost of afforestation in Hong Kong, and the observation that fire protection can promote fairly rapid shrub invasion which proceeds readily to secondary forest, control of hill fires would appear a better option for future forest conservation policies.

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